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LAND USE IN LCA

The use of temporal dynamics for the automatic calculation of land use impacts in LCA using R programming environment

A case study for increased bioenergy production in Luxembourg

Ian Vázquez-Rowe · Antonino Marvuglia · Katja Flammang · Christian Braun · Ulrich Leopold · Enrico Benetto

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Abstract
Purpose Evaluation of soil functionality in Life Cycle Assessment (LCA) has progressively gained importance, although only a small cluster of studies deliver detailed guidelines on how to calculate quantified indicators. In addition, there is a lack of bibliography assessing impacts on the pedosphere due to spatially differentiated land use changes (LUC). In this study, an automated geospatial simulation of LUC based on crop rotation probabilities in Luxembourg was implemented in the programming environment R. Furthermore, this method based on coupling LCA and geographic information system (GIS) was used to calculate changes in soil functionality by implementing both the soil organic carbon (SOC) method and the Land Use Indicator Value Calculation Tool (LANCA®). The developed R script was then applied to a case study dealing with maize production for bioenergy purposes in Luxembourg.
Methods On the one hand, geo-referenced crop information in Luxembourg for the period 2005–2011 was used to calculate the estimated probability in which crop rotation occurs in combination with maize expansion to meet bioenergy production requirements by 2020. On the other hand, geo-referenced information for a wide range of parameters relevant in assessing soil functionality was stored in a geospatial database and mapped using R's geospatial data manipulation, analysis and visualisation capabilities. The geospatial data were used as input for the R LANCA® model, which calculates the environmental impacts associated with the five indicators considered in the model (erosion resistance, physicochemical filtration, mechanical filtration, biotic production and groundwater replenishment) for all the cultivated areas in Luxembourg.
Results and discussion The application of the two models demonstrated the significant differences in soil functionality in Luxembourgish arable land, namely between the north and south of the country. Spatial differentiation was found to be important in all indicators, except biotic production and physicochemical filtration, in which the availability of more detailed datasets and more specific methods is a must. Finally, the coupling of GIS and LCI data proved to be an interesting tool for estimating transition probabilities in crop rotation and, therefore, useful in forecasting suitable areas to implement future agricultural policies.
Conclusions GIS and LCI data coupling may constitute an interesting pathway to combine environmental impact assessment and spatial differentiation, provided that further improvements are performed in the method, including important soil parameters or farmer behaviour. In addition, the spatial mapping of environmental impacts can provide important support in terms of policy-making, conservation of natural resources, landscape, or agricultural planning.

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I. Vázquez-Rowe · A. Marvuglia · K. Flammang · C. Braun · U. Leopold · E. Benetto (✉)
Public Research Centre Henri Tudor (CRPHT)/Resource Centre for Environmental Technologies (CRTE), 6A, avenue des Hauts-Fourneaux, L-4362 Esch-sur-Alzette, Luxembourg
e-mail: enrico.benetto@tudor.lu

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1 Introduction

It has been demonstrated in a wide range of studies that land use changes (LUC) can potentially trigger important effects on environmental impacts, especially greenhouse gas (GHG) emissions (Milà i Canals et al. 2007a; Searchinger et al. 2008; Giampietro and Mayumi 2009; Hertel et al. 2010). Nevertheless, other factors, such as intensification of agriculture or damage on soil functionality, can create important degradations in terms of the quality of the soil (Milà i Canals et al. 2007a; Garrigues et al. 2012). Consequently, environmental management tools, such as Life Cycle Assessment (LCA) (ISO 14040 2006; ISO 14044 2006), have started to integrate specific assessment tools and impact categories to assess land use (Milà i Canals et al. 2007a). However, despite the growing attention gained by the topic, a wide range of LCA case studies still fail to assess the land use environmental profile of their particular production system beyond the quantification of the occupied land use (Milà i Canals et al. 2007c; Villanueva-Rey et al. 2013), ignoring, for instance, the changes in soil organic carbon (SOC) associated with growing biomass in the assessment of bioenergy production (Larson 2006). In addition to SOC, erosion, microbial biomass, salinisation, ecosystem thermodynamics/exergy (measurable indicators), threats to endangered species and ecosystem damage potential (EDP) (Koellner and Scholz 2007, 2008) have been suggested as possible indicators for land use impacts in LCA (Mattila et al. 2012; Brandão et al. 2011). Hence, in recent years, different studies have explored and suggested several pathways to include land use impacts in LCA in terms of qualification (Cowell and Clift 2000; Feitz and Lundie 2002; Wagendorp et al. 2006; Milà i Canals et al. 2007a; 2007b; Beck et al. 2010; Brandão et al. 2011; Mattila et al. 2012).

Thus, while current European Union (EU) policy advocates for the promotion of renewable energies, including the use of biomass, the use of agricultural crops for energy production has demonstrated to be directly related to an increase in climate change, acidification or eutrophication impacts due to variable factors, such as fertilizer runoff or biodiversity loss (Searchinger et al. 2008; Giampietro and Mayumi 2009). However, in many cases, it remains unexplored how these changes in land use due to increasing land areas destined to bioenergy production may affect the quality of the land in a given area. In fact, given the local influence of land use on soil properties, a site-specific analysis of potential land use impacts is intended to reduce uncertainty and unveil local effects which would remain undisclosed by an aggregated approach (Reijnders and Huijbregts 2008; Wicke et al. 2008; Börjesson and Tufvesson 2011; Brandão et al. 2011).

Luxembourg is the smallest nation in the EU after Malta, with a total territory of 2,586 km^2. Its population density is high, experiencing a continuous increase in recent years due to its demographic and economic expansion (EEA 2010). Its strong economic proliferation has also increased the pressure on a wide range of environmental dimensions, such as climate change due to greenhouse gas (GHG) emissions or biodiversity (Jury et al. 2013). Due to limited land availability, land management arises as a major environmental problem. For instance, according to the European Environment Agency (EEA) (2010), agricultural and wetland areas declined by 147 km^2 (5.7 % of total land in Luxembourg) between 1990 and 2009, and are expected to continue to decrease in years to come, triggering the existing pressures on natural resources (EEA 2010; STATEC 2013). Nevertheless, as of 2009, 35.3 % of Luxembourgish territory was woodland, and agricultural area represented 50.6 % of the land, while the transportation network and human settlements only covered 13.5 % of the total land use.

A cluster of studies performed in Luxembourg (Marvuglia et al. 2013; Vázquez-Rowe et al. 2013a; 2013b) assessed from a consequential LCA (C-LCA) perspective the predicted changes in environmental impact linked to the domestic agricultural sector if maize cultivation were to be expanded to produce biomethane in the period 2009–2020. Interestingly, despite the limited increases in environmental impact in terms of conventionally used impact categories (e.g., global warming, eutrophication, acidification, etc.) for the modelled scenarios, an important growth in land use impacts (i.e., using land occupation and transformation approaches of standardised assessment methods, such as ReCiPe or Impact 2002+) was observed, suggesting the need for further research to monitor the qualitative fluctuations that may occur in the soil due to LUC. However, it should be noted that changes in land use were limited to crop rotation rather than larger changes of land use types due to the restrictive nature of agricultural legislation in Luxembourg, impeding shifts in land use types (MAVDR 2005).

Therefore, the main aim of this study was to perform an environmental assessment of land use in the Luxembourgish agricultural system by using two different quantitative approaches: the soil organic carbon (SOC) approach developed by Milà i Canals et al. (2007a) and further developed by Brandão et al. (2011), on the one hand, and the LANCA® model (Beck et al. 2010), on the other. Based on the data obtained for 80 different soil samples on agricultural land in Luxembourg, the main inventory results and methodological constraints of applying these two methods are discussed and compared. Moreover, the calculation process was implemented as a fully automated code in the open source programming environment R (R Development Core Team 2008). The

application of the implemented code allows the automatic processing of the geospatial information, thus permitting: (i) the spatial differentiation of LUC expected in 2020 based on crop transition probabilities estimated using historical land use maps and on the expected crop production in 2020 calculated via an economic model (Rege et al. 2013; Vázquez-Rowe et al. 2013a) and (ii) the automatic calculation of regionalised potential impacts on the soil functionalities associated with the agricultural land use patterns (see Fig. 1). Finally, a set of conclusions are extracted from the results obtained concerning the land use differences throughout the Luxembourgish agricultural sector.

2 Materials and methods

2.1 Background

Following the 20/20/20 EU targets (EU 2009), the government of Luxembourg predicts that by 2020, 11 % of the domestic gross energy demand should be provided by renewable sources linked to bioenergy (i.e., biofuels). More specifically, if the amount of energy that should be covered by biogas production is taken into consideration, this value would sum up to a total of 144 GWh in 2020. The assumption made in this study, which derives from work developed in Vázquez-Rowe et al. (2013a) and Rege et al. (2013), is that this amount of biogas would be supplied domestically through an augmentation in the production of energy maize to produce biomethane.

This assumption leads to a predicted increase in 80,000 t of maize produced for bioenergy purposes in the period 2009–2020. This *shock* on the Luxembourgish agricultural system implies that there will be a series of changes in the crop distribution within the arable land, as demonstrated by Vázquez-Rowe et al. (2013a). The latter publication estimated and discussed the environmental consequences of applying this hypothetical shock, using a partial equilibrium (PE) model in combination with a C-LCA perspective, showing that one of the major sources of increased environmental impacts in the 2020 scenarios would be the impacts linked to land transformation (i.e., the impacts linked to changes in ecosystem quality) and land occupation, which represent the impacts related to the continuation of a conversion that is non-natural (i.e., it does not correspond to the potential natural vegetation) for a particular location (Koellner et al. 2013).

2.2 Goal and scope definition

The main objective of this study is determining the environmental impacts of the Luxembourgish agricultural system in

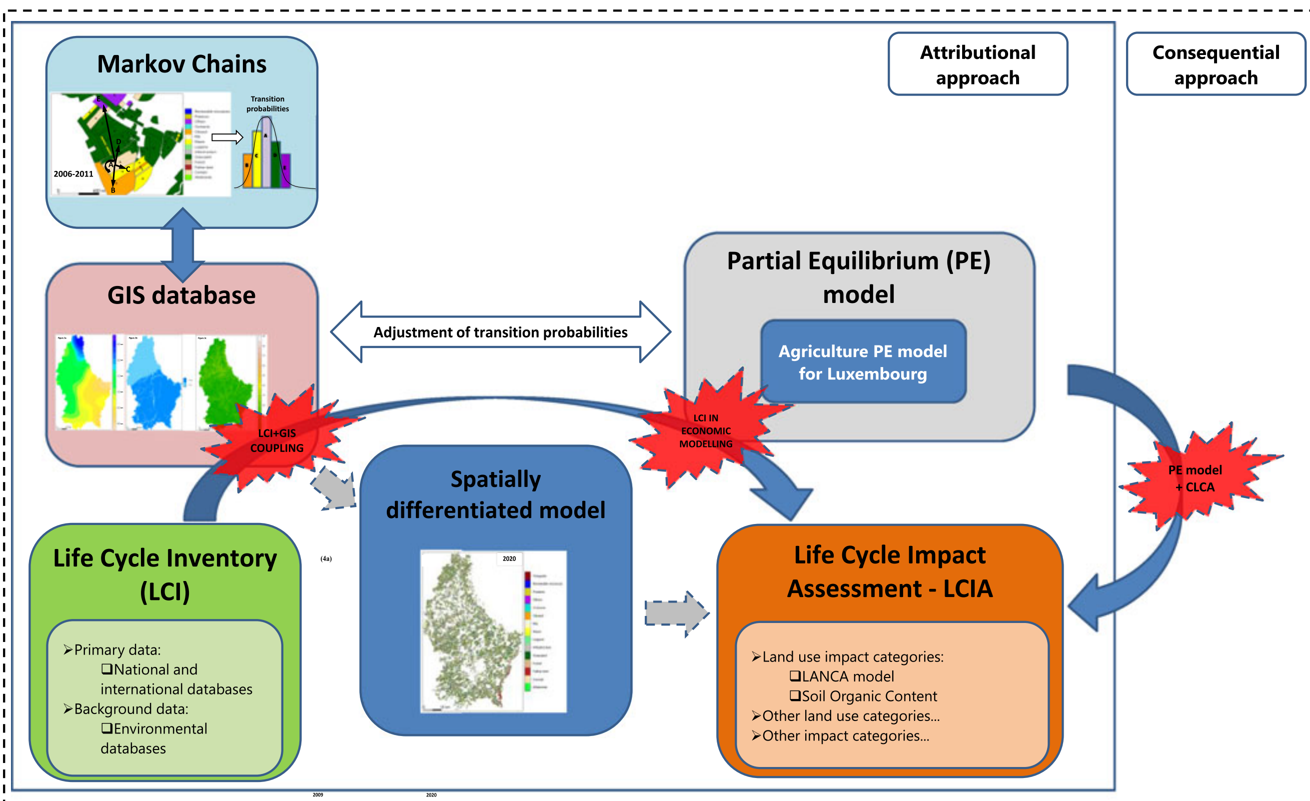

Fig. 1 Methodological approach for the spatial differentiation of land use environmental impacts. The *outer dotted line boundary* represents a consequential Life Cycle Assessment (CLCA) perspective, beyond the scope of the current article. The *inner continuous line* represents the attributional perspective implemented in the current case study

terms of land use impacts, since it has been a repeatedly underexploited impact category in many agricultural LCA studies. Consequently, on the one hand, the spatial differentiation of arable LUC in Luxembourg was modelled by regionalising the land use patterns for the period 2006–2011 and, thereafter, simulating the spatial distribution of crops by 2020 using a Markov chain model assuming a random process (Castellazzi et al. 2010; Sorel et al. 2010). The Markov property states that the conditional probability distribution for the system at the next step and all future steps depends only on the current state of the system, and not on the state of the system at previous steps. These changes of state are called transitions, and the probabilities associated with various state changes are called *transition probabilities*. The set of all states and transition probabilities completely characterises a Markov chain (Davis 2002).

It should be noted that land use environmental impacts were not approachable through the same methodological perspective (i.e., consequential changes due to LUC linked to crop rotation in the 2009–2020 time frame) due to the lack of data available for soil parameter differentiation between agricultural crops, as well as to intrinsic current limitations of the assessment methodologies, which are suited to assess impacts linked to actual LUC, but do not have enough granularity to assess impacts simply linked to crop rotation. The approach employed to monitor these environmental burdens was based on an attributional LCA (ALCA) perspective to calculate the difference in soil quality occurring between the transformation time and the natural state potentially reached by the land after a *relaxation* time (Milà i Canals et al. 2007b). According to the latter, the soil quality is measured in terms of SOC content, which is closely related to many other soil quality indicators such as cation exchange capacity and soil life activity (Mattila et al. 2012). However, the SOC indicator does not cover all aspects of ecological soil quality; it rules out important aspects such as soil erosion, compaction, build-up of toxic substances, acidification, salinisation or the depletion of nutrients and groundwater (Milà i Canals et al. 2007c). Therefore, the LANCA® model was also computed, covering partially some of these aspects and allowing the determination of the potential impacts on the ecological quality of land based on detailed site-specific data.

2.3 Timeline scenarios

Two different scenarios were modelled for the assessment of the LUC, following the typical restoration land use types in NW Europe. These scenarios were based on direct input provided by experts on soil chronosequences in NW Europe (Dr. Didier Stilmant and Dr. Bas van Wesemael, personal communications, 2012). *Scenario A* considers that after the occupation of the land in 2009, a natural relaxation time follows which brings to a *potential* state after relaxation (Lindeijer et al. 2002), reached in 2029, identified as grassland. *Scenario B*, following the same rationale, assumes that by 2109 the potential state after relaxation is deciduous forest (see "Section 2.6"). The choice of this two-step process allows the calculation of the intermediate state of the soil when land occupation ends and the land evolves towards the state identified as its *relaxation potential* (Milà i Canals 2003). Chronosequential analysis of soil development has shown to be a recommended tool for monitoring those characteristics linked to pedology that present relatively predictive evolutions, such as soil organic matter, plant cover or pedogenesis (Walker et al. 2010; Huggett 1998). Hence, the use of this approach was deemed suitable for the selected LCA perspective.

2.4 Function and functional unit

The function of the system under analysis is two-fold. On the one hand, the predicted land use patterns in Luxembourg by 2020 were determined based on inter-crop transition probabilities and on the predictions obtained using a partial equilibrium (PE) model (Vázquez-Rowe et al. 2013a). For this perspective, the selected functional unit (FU) was the entire Luxembourgish arable land surface. On the other hand, the land use impacts were calculated through temporal dynamics of selected soil parameters. In this case, the chosen FU was 1 ha of agricultural land, since most of the acquired data used in the assessment methods was readily available for this surface unit. This FU was applied to the 80 sample points available throughout the arable land, extrapolating the values to 1 ha. In addition, the results were reported in terms of the environmental changes between the two timeline scenarios, as explained in "Section 2.3", and the reference year (i.e., 2009).

Other FU approaches (i.e., mass or energy) were disregarded due to the lack of disaggregation between soil types based on the crop cover, as mentioned in "Section 2.1", which only allows the assessment in terms of land use type change (i.e., forest or grassland). Nevertheless, a second stage embraced the scaling up of the environmental impact results to the entire country by spatially mapping the soil-related input parameters (soil texture, precipitation, groundwater distance…) used to calculate the land use impact indicators and, thereafter, automatically compute the values of the indicators using the ad hoc R code mentioned above.

2.5 System description and boundaries

The total area destined to agricultural production in Luxembourg summed up to 130,762 ha in 2009, representing approximately 50.6 % of the territory. Most of this land is occupied by pastures (44.6 %) and cereals (23.3 %). However, maize, which occupies 12.3 % of the arable land in Luxembourg, is becoming an important source for

biomethane production for biofuels (Arrouays et al. 2002). Consequently, the use of maize and other crops for the production of bioenergy constitutes an attractive option for local authorities, as a way of reducing the energy dependence and increasing the revenues of the primary sector.

The system boundaries were limited to the agricultural phase of the different crops, meadows and pastures that were included within the Luxembourgish agricultural system. More specifically, the boundaries included only the soil column of arable land in order to calculate the selected environmental indicators for land use impacts.

2.6 Data acquisition and life cycle inventory

A total of 80 different soil samples were retrieved from a soil database for Luxembourg (Schanne and Mathieu 2006; see Fig. 2 and Tables S1 to S5 in the Electronic Supplementary Material). These samples included a wide range of parameters for soil characterisation, such as their geographical location, number of horizons, sand, silt and clay content, type of soil, humus and skeletal content, pH, bulk density, depth of soil per horizon or organic carbon content. The latter item was the only parameter used, besides the location of the samples, to compute the SOC category. The others, however, were used to compute the results for the five parameters included in the LANCA® model.

Moreover, geo-referenced information for several parameters relevant in assessing soil functionality, such as precipitation and evapotranspiration, soil properties, land use class or declination, are stored in a geospatial database and were mapped using R's geospatial data manipulation, analysis and visualisation capabilities (Fig. 3).

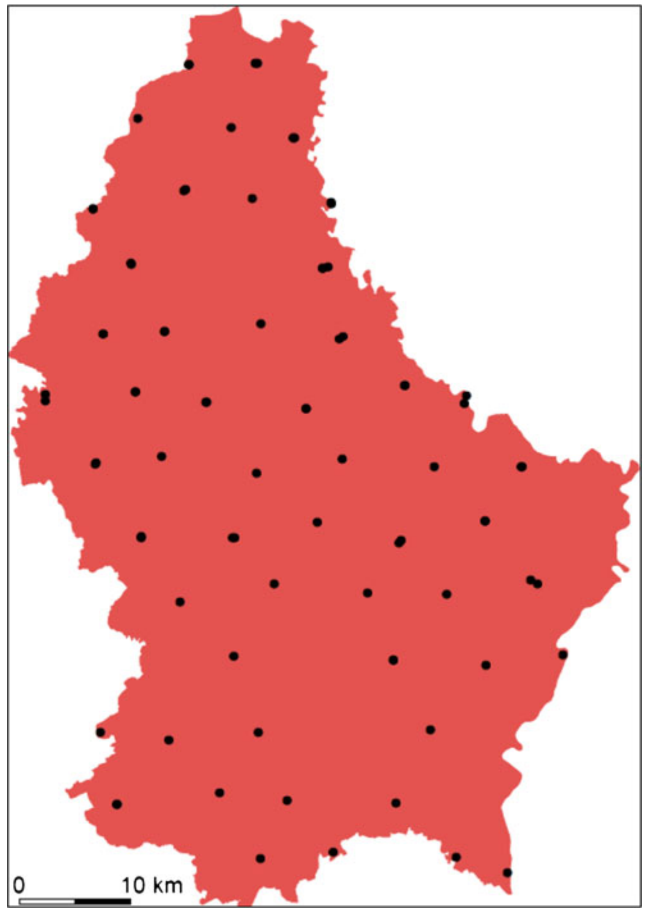

Fig. 2 Map of Luxembourg representing the location of the 80 soil sample points

2.7 Life Cycle Impact Assessment (LCIA) methods

As mentioned above, two different methods were applied to evaluate the potential impacts on soil quality arising from LUC. The selection of these two assessment methods (i.e., LANCA® model and soil organic carbon (SOC)) was based on the fact that they constituted the two models for which the available inventory data fitted in correctly, allowing the computation of all indicators. On the one hand, the LANCA® model, which adapts the methods developed by Baitz et al. (2002), evaluates the impact of land use on soil functionalities accounting for five different indicators: erosion resistance (ER), mechanical filtration (MF), physicochemical filtration (PF), groundwater replenishment (GR) and biotic production (BP) (Beck et al. 2010). It is based on soil parameters which vary at a small, site-specific scale, for which even an approach based on site-dependent characterisation factors is not appropriate (Koellner et al. 2013).

On the other hand, the soil organic carbon (SOC) indicator, developed by Milà i Canals et al. (2007a) was calculated for each sample point, using the revised method suggested by Brandão et al. (2011) as follows:

$$\Delta C\left[\frac{t\ C\ yr}{ha\ yr}\right] = \frac{\left(SOC_{pot} - SOC_{ini}\right) \times \left(t_{relax} - t_{ini}\right) + 1/2\left(t_{relax} - t_{ini}\right) \times \left(SOC_{ini} - SOC_{fin}\right)}{\left(t_{fin} - t_{ini}\right)} \quad (1)$$

where SOC_{pot} is the predicted level of carbon content reached by the soil in the steady state it attains when left undisturbed, SOC_{ini} is the SOC level at the start of the land use studied, SOC_{fin} is the SOC level at the end of the cultivation period, t_{ini} is the time when the studied land use starts and t_{fin} is the time when it ends, and t_{relax} is the time when the soil quality has reverted (via natural relaxation occurring after abandonment of the land) to its level prior to LUC. It has to be noted that

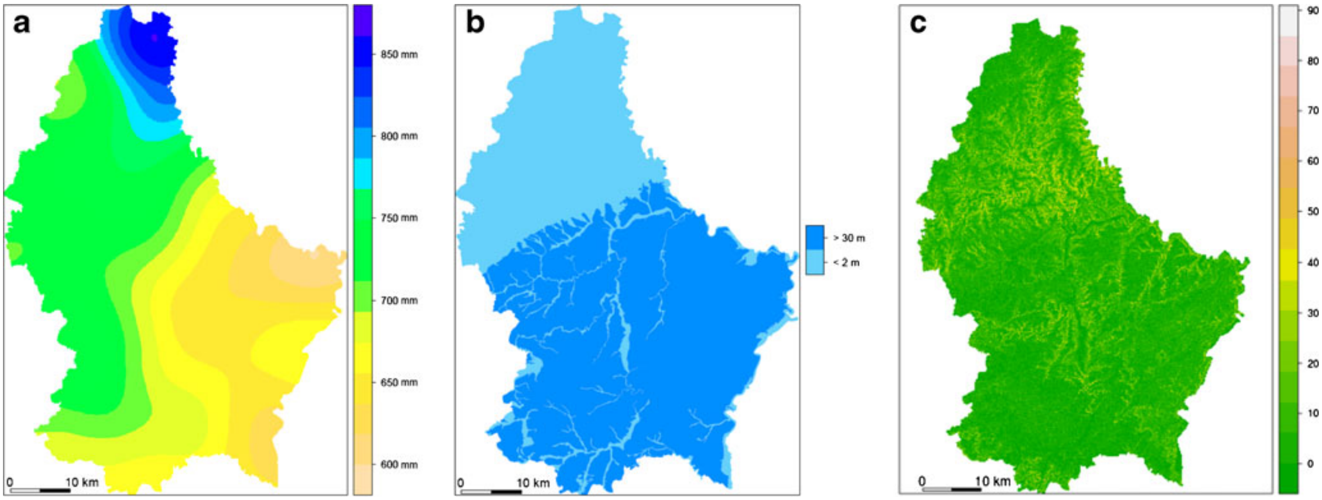

Fig. 3 Spatially differentiated representation of of precipitation (**a**), groundwater distance (**b**) and slope (**c**) in Luxembourg

SOC_{pot} (which is called relaxation potential[1]) is in many cases lower than the SOC in a natural (pristine) situation, before human-induced LUC started, because the re-naturalisation process may well be incomplete (i.e., the impact produced by the LUC is not reversed). In this case, a permanent change in soil quality is produced, which is called *transformation impact* (Milà i Canals 2003).

Since the samples had been collected through boreholes capturing deep soil samples divided by horizons, SOC was calculated following Eq. 6 in Milà i Canals et al. (2007b), which allows the calculation for a generic sample with n soil horizons. It is reported here for the sake of completeness as follows:

$$SOC_{total} = \sum_{i=1}^{n} (\%C_i \times Bulk\ density_i \times Horizon\ tickness_i) \tag{2}$$

where C_i is the soil organic matter concentration (in percentage) of the ith soil horizon.

The thickness of each sample varied from 20 to 40 cm, depending on the depth of the surface soil[2] in each sample point. Therefore, based on Eq. 1 and on the data from the database, it was possible to model the SOC level at the beginning of the assessed land uses (SOC_{ini}). Therefore, despite the fact that the SOC indicator proposed by Milà i Canals et al. (2007a) assumes that only the total organic carbon in the top 30 cm of soil should be accounted for, since it is up to this depth that major changes in carbon content can occur due to land management, for this case study, a flexible distance was considered based on the thickness of the surface soil. SOC_{pot} was modelled based on the data provided by Poeplau et al. (2011), as explained in "Section 2.6". Finally, SOC_{fin} was calculated based on a relaxation rate of 0.32 t C ha^{-1} year^{-1} (Brandão et al. 2011).

2.8 Crop simulation models using a Markov chain approach

As mentioned above, one of the aims of this study is to produce a land use map for 2020, based on the probability of crop transition inferred from historical land use maps from 2006 to 2011. More precisely, the ultimate aim of the simulation is to confer a spatial dimension to the crop patterns for 2020 predicted by using the PE model described in Rege et al. (2013). In particular, the PE model calculates the hectares of land occupied by the different crops in 2020, considering farmers' revenue maximisation and imposing an additional amount of 80,000 t of maize which has to be produced domestically. This quantity, which represents the shock imposed to the agricultural system in the modelling exercise, represents the amount of maize necessary to meet the target set, in terms of biogas production, by the Luxembourg's Renewable Energy Action Plan (LUREAP) (Ministére de l'Economie et du Commerce Extérieur 2012). However, the PE model is only able to calculate the expected surface areas for the different crops, without providing any information about their spatial location. Thus, an actual arable land use map for 2020 would not be possible without a spatially based approach, which is the aim of the crop simulation carried out through the Markov chain algorithm.

The crop type of parcels with crop data available for two consecutive years was listed in a so-called *transition matrix* (if for 1 year there was no crop data—appearing or disappearing parcels—then the parcel was excluded for these 2 years). Therefore, the matrix contained the crop rotations for 2-year intervals (2006–2007, 2007–2008 and so forth until 2010–2011). For each crop type, c_i the frequency of occurrence of

[1] In this approach suggested by Brandão et al. (2011), the relaxation potential (thus, neither the pristine, natural state, nor the state existing at the time when the assessed land use occurred) is taken as the reference state for calculation of the *occupation impact*.

[2] Surface soil refers to the soil horizons that are situated in the upper section of the soil column and hosts the larger portion of organic matter accumulation and soil life (FAO 1998).

each other crop type c_j ($j=1\ldots N$) as the destination crop of c_i (i.e., the crop c_i turned into) was calculated over all the years. These frequencies were collected in the transition matrix and used as estimates for the transition probability (see Table 1).

The prediction of the crop patterns for the subsequent year (e.g., 2012) was carried out by selecting a parcel from 2011, allowing the land use map of 2011 to read the crop type of that parcel and attributing a crop type to the parcel based on the transition matrix. This operation was repeated for all the parcels in order to have the complete crops map for 2012. The prediction for the following years was obtained analogously, using each time the crop type determined for the preceding year (e.g., 2012) to predict the crop type in the following year (e.g., 2013).

Once the map for 2020 was obtained, crop types were aggregated into crop classes, since the crop types contained in the map (as in Table 1) did not coincide with the crops differentiation used by the PE model (which included only 21 crops as opposed to 24 crop types). In order to match them, the individual crops of the map were aggregated into four classes: maize, cereals, oilseeds and grassland. Finally, the transition probabilities shown in Table 1 were adapted in order to match the amount of hectares of each of these four crop types predicted by the PE model, accounting for the future developments of the agricultural economic system (Rege et al. 2013). However, it should be noted that it was inevitable to have a certain degree of discrepancy between the surface area per crop type in 2020 determined using a geographic information system (GIS)-based high resolution approach for the transition probabilities and the amount of surface for each crop originating from aggregated national statistics determined for 2020 by the PE model (see Table 2). As a result, for the 2020 predictions, the transition probabilities were manually adjusted in the matrix to match the PE model predictions.[3]

2.9 Assumptions

For the Milà i Canals model, the current SOC was available for the different samples. However, there was no sampling information or any regional factors for the temporal dynamics in SOC linked to LUC. Therefore, the relative SOC changes after land use conversions in the temperate zone were extracted from Poeplau et al. (2011), as can be seen in Table 3, based on carbon response functions (CRFs).

Regarding the evolution of land quality over time, it is important to note that only two different time-dependent steps were considered. Firstly, arable land in Luxembourg has been roughly stable in recent decades, with the major changes being at the level of crop rotation. Given that the available soil samples were only reported for one single time period (during occupation), it was assumed that land quality was constant not only throughout the occupation (Beck et al. 2010) but also at the beginning of the land use (i.e., cropland or grassland), since the land use type is constant. Therefore, no differentiation is included between the initial state of the analysed surface, the start of the occupation and the final occupation, impeding the monitoring of quality changes throughout this period. Secondly, the relaxation time until the quality of the soil is restored (or partially restored), which eventually measures the transformation impacts, was modelled based on the SOC predictions described in "Section 2.6", and by assuming two future scenarios of LUC (described in "Section 2.2").

3 Results

3.1 Crop rotation pattern changes

Once the arable land maps for Luxembourg in the period ranging from 2006 to 2011 were completed (Fig. 4a shows the crop distribution for 2009), the transition probabilities were computed for these historical time series. Thereafter, these probabilities were used in a Markov chain approach to predict the annual business-as-usual crop rotations up to 2020. In order to harmonise the crop rotation distribution (shown in Table 4) estimated by the PE model, as described in Vázquez-Rowe et al. (2013a) and Rege et al. (2013), the transition probabilities were manually optimised to match the PE model results for 2020. Figure 4b presents the predicted distribution of crop patterns in 2020. Figures 5a and 5b present a detail of the respective maps shown in Figs. 4a and 4b.

As may be observed in Fig. 4b, the amount of land destined to maize increases considerably in the selected time period. Moreover, the prediction of crop type results becomes more identically distributed through time.

3.2 LANCA® model results

3.2.1 Erosion resistance

ER can be defined as the capability of a specific unit of land to resist erosion episodes beyond the naturally occurring erosion processes (Beck et al. 2010). This capability is strongly influenced by the slope and texture in a given point (Bastian and Schreiber 1994). Tables S11 to S14 in the Supplementary Material show the transformation and occupation midpoint values for the 80 sample points analysed. As can be seen in Fig. 6 (see also Tables S1 and S6 in the Electronic Supplementary Material), the areas of arable land in Luxembourg with a higher resistance to erosion in 2009 were those in the north of the country. These results are mainly

[3] The use of an optimisation algorithm or an inverse solution was not implemented to adjust the transition probabilities in the matrix to the PE model since this would have been beyond the scope of this study.

Table 1 Adjusted transiton matrix (% rounded) for the analysed production system. The value in the position (*I*,*J*) of the table represents the probability that the crop indicated as the header of row *i* will turn into the crop indicated as the header of column *j*. Note: values in bold represent the adjusted probabilities, i.e., divergence from business as usual to match the target values of 2020 computed by the partial equilibrium (PE) model

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T	U	V	W	X
A	8.18	5.56	0.05	0.00	0.61	0.00	0	0.03	0.00	0.00	1.93	0.00	0.13	0.00	0.31	0.69	2.44	0.00	0.11	2.72	0.00	7.13	0.00	0
B	0.00	16.67	0.00	0.10	0.00	0.00	0	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0
C	7.43	11.11	47.11	15.10	20.01	20.92	0	**0.97**	5.06	14.29	34.91	27.81	**49.69**	28.46	69.23	39.13	5.72	4.72	12.44	52.98	42.01	9.36	2.56	0
D	0.00	0.00	0.39	23.08	0.24	6.35	0	0.18	6.46	14.29	0.38	9.93	0.18	9.33	0.32	16.70	0.00	0.00	0.51	0.40	3.20	0.15	2.56	0
E	8.55	0.00	0.62	0.00	34.67	1.88	0	0.15	0.00	0.00	1.55	0.00	0.32	0.20	0.52	0.00	3.15	0.00	0.53	2.72	0.26	8.77	0.00	0
F	0.00	5.56	0.12	1.07	0.53	20.20	0	0.03	0.42	0.00	0.10	0.00	0.04	1.20	0.10	1.83	0.06	0.00	0.09	0.17	0.52	0.00	0.00	0
G	0.00	0.00	0.00	0.00	0.00	0.00	70	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0
H	16.36	27.78	**2.38**	22.42	15.43	12.99	0	90.50	24.16	42.86	11.57	5.96	**6.89**	16.77	12.23	10.53	22.48	23.11	14.46	10.27	11.41	15.16	15.38	0
I	0.00	0.00	0.06	1.98	0.00	2.02	0	0.10	45.37	0.00	0.10	1.99	0.04	0.65	0.04	0.46	0.00	1.89	0.23	0.00	0.17	0.00	0.00	0
J	0.00	0.00	0.00	0.00	0.00	0.00	0	0.00	0.00	28.57	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0
K	8.55	0.00	1.11	0.36	1.42	0.43	0	0.13	0.14	0.00	25.23	9.93	0.39	0.24	0.72	0.23	2.63	0.47	0.48	2.55	0.61	7.43	2.56	0
L	0.00	0.00	0.06	0.66	0.00	0.14	0	0.01	0.00	0.00	0.14	19.21	0.03	0.48	0.06	0.46	0.00	0.00	0.02	0.00	0.61	0.00	0.00	0
M	7.43	16.67	**27.55**	4.93	6.57	6.93	15	**4.80**	1.83	0.00	6.68	6.62	36.05	15.37	8.24	7.55	3.66	5.19	4.50	5.96	4.58	8.77	5.13	0
N	0.00	11.11	0.88	9.30	0.77	5.63	5	0.2	4.21	0.00	0.45	4.64	1.04	20.26	0.75	5.49	0.00	0.00	0.46	0.40	3.37	0.15	10.26	0
O	9.29	0.00	11.34	1.88	2.30	3.32	0	0.58	0.98	0.00	3.58	4.64	1.04	1.02	2.37	2.75	3.79	0.47	0.92	3.23	2.42	6.39	0.00	0
P	0.00	0.00	0.20	6.20	0.02	1.88	0	0.02	0.42	0.00	0.10	0.00	0.04	0.99	0.11	7.32	0.00	0.00	0.04	0.06	0.86	0.00	0.00	0
Q	6.51	0.00	0.11	0.15	1.07	0.00	0	0.13	0.14	0.00	1.27	0.00	0.20	0.00	0.43	0.00	45.66	7.08	0.15	2.21	0.00	6.54	0.00	0
R	0.00	0.00	0.01	0.00	0.00	0.58	0	0.03	0.42	0.00	0.00	0.00	0.02	0.07	0.01	0.00	0.39	55.66	0.00	0.00	0.00	0.15	0.00	0
S	10.78	0.00	6.57	10.47	13.78	13.85	5	1.95	9.27	0.00	8.06	5.96	3.12	3.18	3.20	5.03	4.37	0.94	64.13	8.62	9.94	7.13	5.13	0
T	7.81	0.00	0.81	0.20	1.27	0.14	0	0.08	0.00	0.00	2.20	2.65	0.46	0.31	0.69	0.69	1.93	0.00	0.57	4.48	0.69	5.20	0.00	0
U	0.00	5.56	0.52	2.08	0.15	2.45	0	0.04	0.70	0.00	0.24	0.66	0.16	1.20	0.34	0.69	0.00	0.00	0.22	0.51	19.36	0.00	0.00	0
V	9.11	0.00	0.09	0.00	1.12	0.00	0	0.05	0.00	0.00	1.45	0.00	0.14	0.07	0.31	0.46	3.73	0.47	0.14	2.72	0.00	17.53	2.56	0
W	0.00	0.00	0.01	0.00	0.02	0.29	0	0.00	0.42	0.00	0.07	0.00	0.01	0.20	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.15	53.85	0
X	0.00	0.00	0.00	0.00	0.00	0.00	0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	100

A allotments, *B* allotments mix, *C* cereals, *D* cereals mix, *E* fallow land, *F* fallow land mix, *G* forest, *H* grassland, *I* grassland mix, *J* infrastructure, *K* legume, *L* legume mix, *M* maize, *N* maize mix, *O* oilseeds, *P* oilseeds mix, *Q* orchards, *R* orchards mix, *S* others, *T* potatoes, *U* potatoes mix, *V* renewable resources, *W* renewable resources mix, *X* vineyards

Table 2 Estimation of surface area per type of crop in 2020 (data in hectares)

Crop type	PE model values 2020 (ha)	Adjusted prediction 2020 (ha)
Maize	22,355	19,729
Oilseed	3,703	4,174
Grassland	59,018	60,652
Cereals	41,830	30,536

linked to the soil texture class, dominated in these areas by silty clay loam, as well as moderate to high humus and skeletal contents. However, varying slopes can also account for some of the variations that distort the pattern between northern and southern soils.

When the midpoint results are analysed for the two modelled scenarios, annual results appear to be similar, with transformation impacts being favourable for the conversion of arable land into grassland and, eventually, deciduous forest. In fact, the provided results may constitute an interesting starting point for policy support in order to identify which areas would benefit from LUC to avoid high episodes of erosion activity (González-Hidalgo et al. 2012).

3.2.2 Physicochemical filtration

This parameter provides information regarding the potential of a specific soil to absorb diluted substances (Bastian and Schreiber 1994). This capability depends strongly on the clay and organic matter content of the soil samples (Bastian and Schreiber 1994; Mückenhausen 1985). However, no specific data were retrieved for different areas of Luxembourg. Therefore, default data for Central Europe were used for the different biomes. This average value for Central Europe was 17.1 cmol $kg^{-1}{}_{soil}$ for cropland and forest, and slightly higher (18 cmol $kg^{-1}{}_{soil}$) for grassland (see Tables S2 and S7 in the Supplementary Material). Figure 7 shows the transformation and occupation impacts for the two modelled scenarios. Scenario A presents a slightly increasing tendency in terms of the cation exchange capacity (CEC) value, while Scenario B shows a stable trend.

Table 3 Soil organic change rates after 20 and 100 years. (Adapted from Poeplau et al. 2011)

Land use changes (LUC)	Minimum estimated equilibrium time	ΔSOC for 20 years (%) (Mg ha^{-1} ± SD)	ΔSOC for 100 years (%) (Mg ha^{-1} ± SD)
Cropland to grassland	120 years	39.8	–
Cropland to forest	120 years	–	83.4±38.8

SOC soil organic content, SD standard deviation

3.2.3 Mechanical filtration

MF accounts for the capacity of the soil to clarify suspended materials, namely pollutants, in the filtered water (Marks 1989). Sandy soils usually have a high filtering capacity, while loamy or silty areas have a reduced filtration potential. Figure 8 shows how the sandy soils in the central and SE of Luxembourg have the highest rates. In contrast, the silty and loamy areas of the rest of the country show less potential for filtration. Finally, regarding the changes in mechanical filtration due to land restoration, no significant differences can be observed between the scenarios and the values calculated for 2009 (see Tables S111–S14 in the Electronic Supplementary Material).

3.2.4 Biotic production

BP can be defined as the amount of biomass [gram per square meter per year] that is not produced in a particular unit of land due to the effects linked to the selected production system (Bos 2010). The results of this indicator, however, are strongly conditioned by the disaggregation of terrestrial biomes included in the LANCA® model. Hence, all the arable land in Luxembourg was considered under the *cropland* or *grassland* biomes, increasing considerably the uncertainties and hampering the usability of this indicator. Figure 9, as well as Tables S4 and S9 in the Electronic Supplementary Material, show the results obtained for BP. The lack of disaggregation between crops and soil types conditions the results strongly. Therefore, only three different values of net primary productivity (NPP) were considered: 650 g/(m^2/a) for regular cropland, 500 g/(m^2/a) for grassland and 650 g/(m^2/a) for deciduous forest. Hence, in terms of quality changes in transformation and occupation (see Fig. 9 and Tables S11–S14 in the Electronic Supplementary Material), BP would show an initial decrease in quality through the relaxation time (Scenario A), and with a slow recovery whenever the land becomes forest (Scenario B). Nevertheless, unless forests with a higher level of biodiversity (e.g., mixed tree woodlands) arise, this indicator would not suffer any substantial improvements after restoration.

3.2.5 Groundwater replenishment

GR is the potential a specific surface of land has to replenish its groundwater resources as a consequence of a wide range of factors, such as vegetation structure, permeability of the soil column, a set of climatic conditions, namely annual precipitation and evapotranspiration, or the distance from the surface to the groundwater (Marks 1989). In Luxembourg, the latter parameter shows a clear division in two areas: a northern area in which the distance to groundwater is 1.5 m, and a central and southern area where the distance to groundwater is above 31 m. This value constitutes, together with the permeability of the soil (which is strongly influenced by the soil texture class

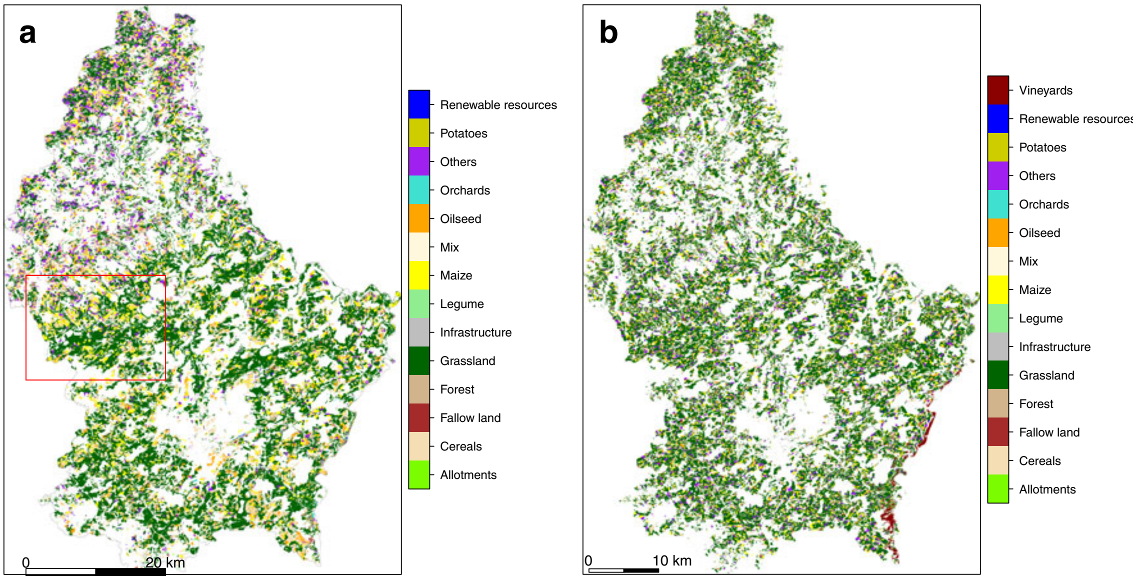

Fig. 4 Arable land use map for Luxembourg in 2009 (**a**); arable land use map for Luxembourg in 2020 based on transition probabilities (**b**)

of each soil), the main parameter that influences the results for this midpoint indicator, since climatic conditions (i.e., precipitation and evapotranspiration) show relatively constant values throughout the entire country, although there is a higher average precipitation in northern areas (see Fig. 3a).

Transformation values, as observed in Fig. 10 and Tables S11–S14 of the Electronic Supplementary Material, show that the soils situated in the north of the country would highly benefit from the transformation process in both scenarios, while the values obtained for the central and southern regions demonstrate a lower potential for improving their replenishment rates.

3.3 Milà i Canals model

The results for the SOC indicator can be observed in Fig. 11 and Table S15 of the Electronic Supplementary Material.

Table 4 Changes in arable land patterns per crop class in Luxembourg in the period 2009–2020 (adpated from Vázquez-Rowe et al. 2013a)

Crop	Surface area 2009 (ha)	Surface area 2020 (ha)	Land use changes (LUC ha)
Cereals	38,446	41,829.86	+3,383.86
Maize	16,488	22,355.43	+5,867.43
Oilseeds	4,629	3,703.20	−925.80
Grassland	67,343	59,017.51	−8,325.49

The original results, available in Vázquez-Rowe et al. (2013a; 2013b) and based on the partial equilibrium model presented in Rege et al. (2013), are disaggregated for individual crops

Figure 11a shows the sample point results for Scenario A, while Fig. 11b presents the sample point values for the conversion into deciduous forest by 2109 (Scenario B). No major spatial pattern can be found in the results, although arable areas in NW Luxembourg were those with a higher potential to store organic carbon in their soil column. On the contrary, areas mainly in SE and central Luxembourg presented the lowest potential increases in SOC during the relaxation periods.

4 Discussion

4.1 The importance of transition probabilities in predicting crop rotation distribution

The results presented in this study aim at providing geospatial details concerning the estimated future scenarios linked to crop rotation in Luxembourg. Moreover, a second step of the research framework was to link the environmental impacts related to LUC on soil functionality through the SOC indicator and the LANCA® model. However, this latter objective was not reachable through the transition probabilities calculated for 2020. This was not only due to the insufficient level of detail of available data but also to the fact that current indicators and impact categories that monitor soil functionality are based on global LUC, that is, changes in land use type, rather than on crop rotation, which changes are considered minor.

Luxembourg is a country in which changes in land use type are strongly monitored and limited by environmental laws

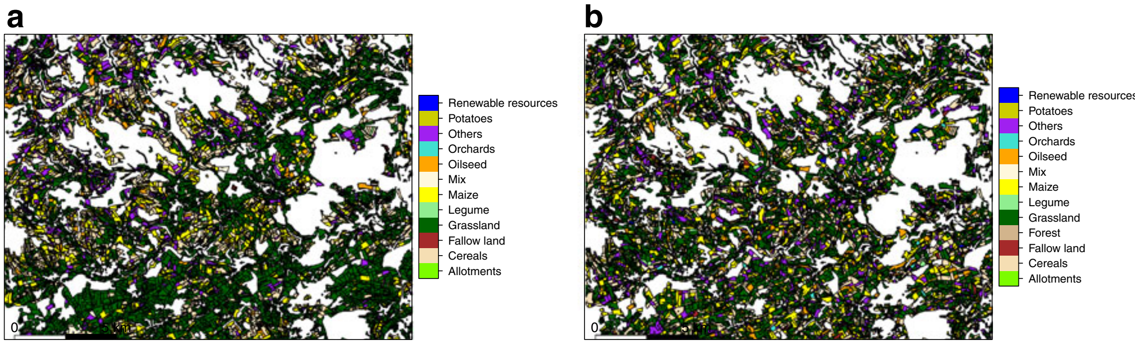

Fig. 5 Detail of Figs. 4a and b. The *zoom out* is taken in the rectangular box superposed to the map shown in Fig. 4a

(SER 2005; Ministry of Agriculture 2009). Therefore, domestic changes in LUC in Luxembourg are going to be linked overwhelmingly to changes in crop rotation, regardless of the indirect LUC that may occur beyond Luxembourgish borders (Vázquez-Rowe et al. 2013a; 2013b). Furthermore, the environmental consequences of these changes through different impact categories in many cases quantify global potential environmental impacts (e.g., climate change or ozone depletion). However, in the case of soil functionality, the environmental changes are always strongly dependent on the geographical variance of biophysical characteristics, e.g., soil properties, climate or hydrology (Geyer et al. 2010a).

Consequently, the changes in crops distribution shown in Fig. 4 in the 2009–2020 time frame were achieved through the statistical analysis of average crop rotations of historical data in the period 2006–2011. In fact, the determined transition probabilities were used to predict future states of crop distribution for each agricultural parcel in Luxembourg on an annual basis using the Markov chain approach. The novelty in this specific study was the integrative perspective of GIS and economic (i.e., PE model) modelling to achieve the spatial mapping of crop rotations predicted through a forward-looking LCA model (see Vázquez-Rowe et al. (2013a) and Rege et al. (2013) for more details). This was attained through manually optimising transition probabilities for certain crops in order to match the outcomes of the PE model.

In addition, the coupling of specific LCI data and GIS, in a similar methodological framework as previous studies (Azapagic et al. 2007; Geyer et al. 2010a; 2010b; Núñez et al. 2010; Mutel et al. 2012) also constituted an important achievement within the aims of the study. While, as mentioned above, the combination of LCI data and GIS seeking the spatial quantification of environmental impacts associated with the pedology of arable land was not feasible within the

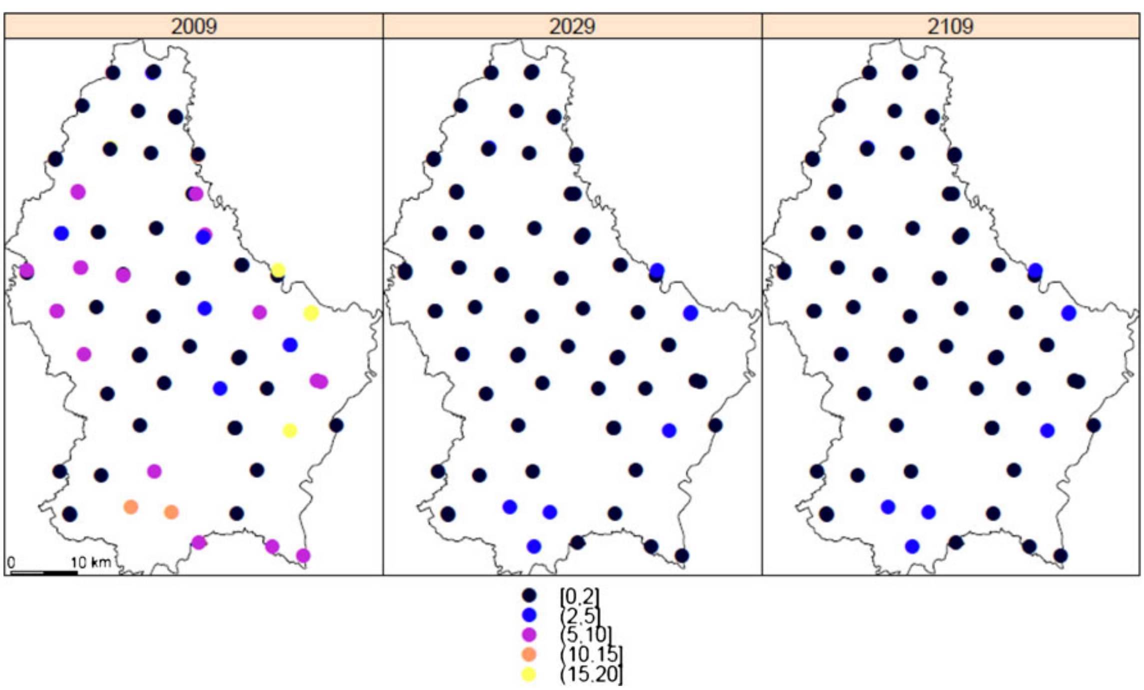

Fig. 6 Erosion resistance (ER) quality values for the scenarios selected. Note: results reported in t/(ha*a)

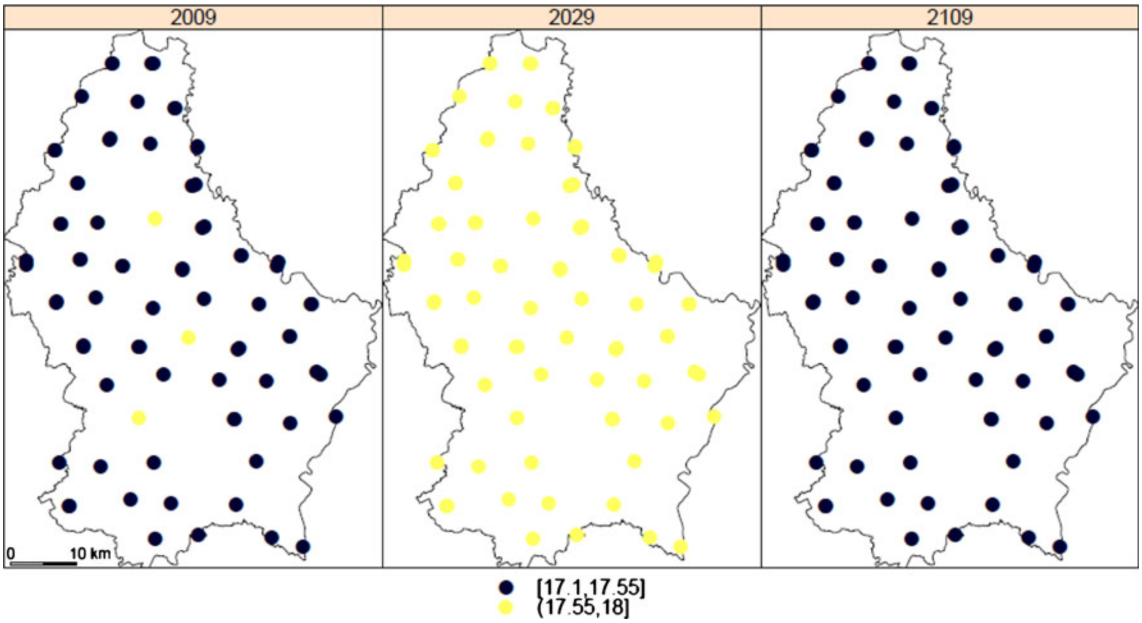

Fig. 7 Physicochemical filtration (PF) quality values for the scenarios selected. Note: results reported in cm/d

period 2009–2020, this coupling provides a useful framework towards determining the feasibility of estimated future scenarios in life cycle thinking. For instance, in this particular case, the coupling allows the visualisation of the main areas where maize production would expand, substituting other crops.

Nevertheless, the methodological framework of this coupling is not absent of a series of limitations and constraints. In the first place, and despite the wide range of spatially explicit data and sample points available, the sampling procedure may be improved if more than the 80 sampling points in the entire country (more details on the methodology are available in "Section S1" of the Supplementary Material) were available. In fact, this issue may be an important source of uncertainty when it comes to map the data spatially. Secondly, another source of uncertainty when providing the spatial mapping was the fact that 40 % of the crop polygons in Luxembourg lacked a link to a specific crop. However, these polygons were, in most cases, smaller polygons that were surrounded entirely by

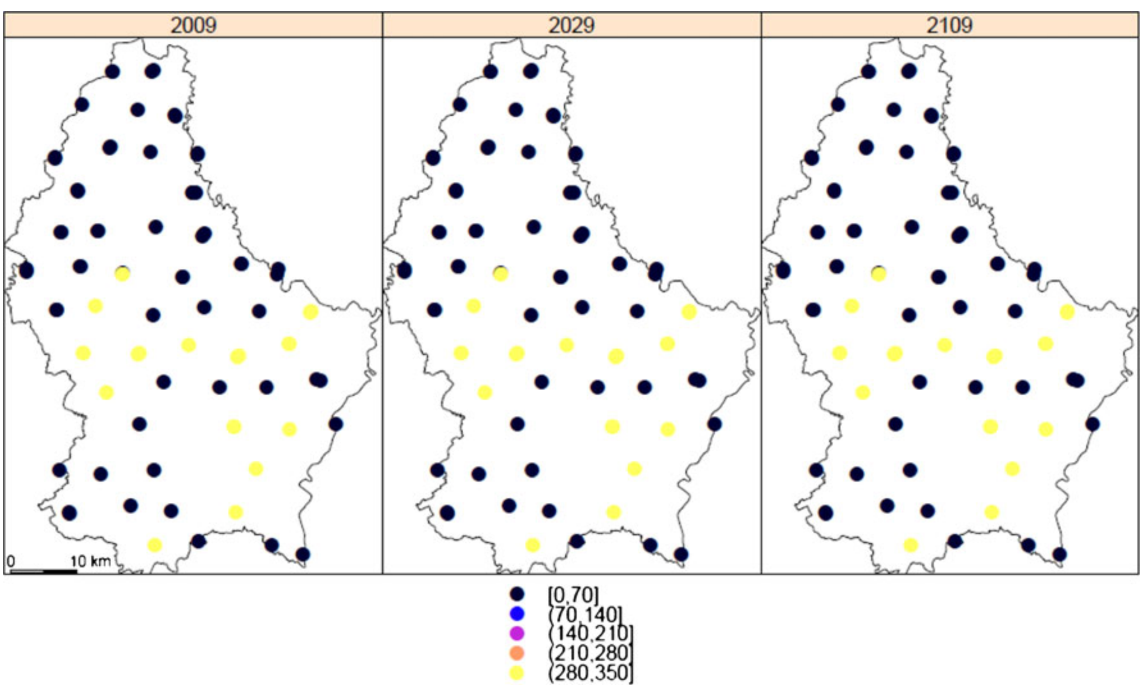

Fig. 8 Mechanical filtration (MF) quality values for the scenarios selected. Note: results reported in cmol/kg$_{soil}$

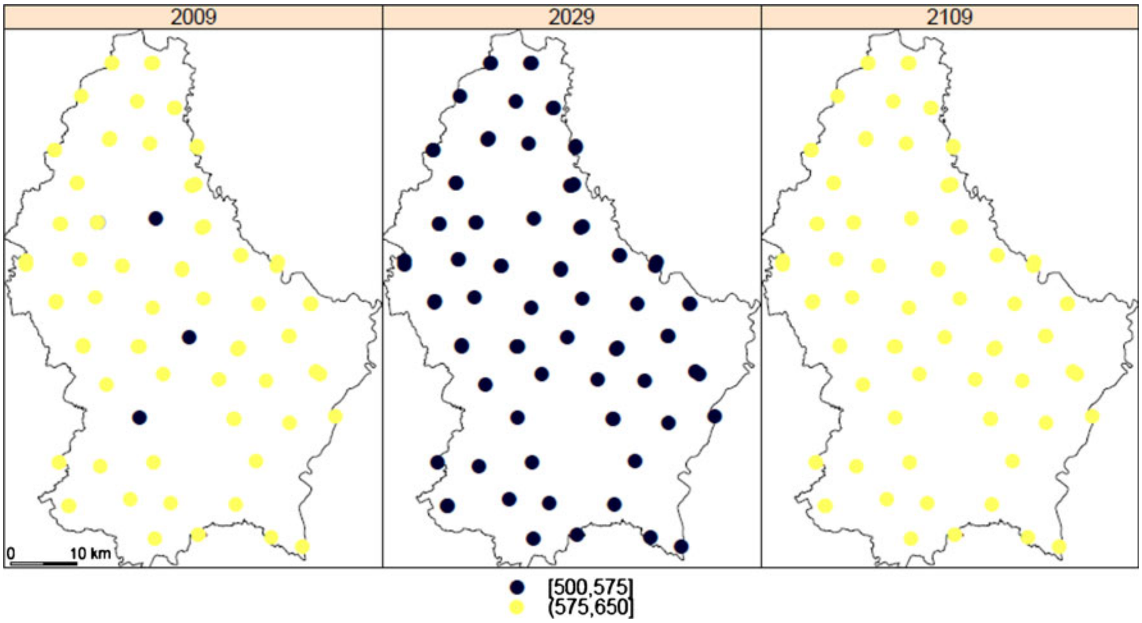

Fig. 9 Biotic production (BP) quality values for the scenarios selected. Note: results reported in g/(m^2*a)

one single crop and represented a small amount of the total arable land in terms of occupied area.[4] Finally, the transition probabilities that were estimated to predict future scenarios were based solely on calculating the frequency throughout the monitored time series (2006–2011) in which a specific polygon changes its crop land use to another. Behavioural aspects of the farmers, as well as the influence of soil characteristics or climate, on the final decision regarding crop rotation or substitution are not accounted for in the coupling model and constitute a major challenge for future improvement of the methodology.

4.2 Brief interpretation of the LANCA® model results

Erosion is an important parameter to be included in life cycle thinking studies, since high episodes of this phenomenon can have shattering effects in terms of the productivity of the land. Moreover, high episodes of erosion have been identified with the production of crops as sources of renewable energy due to intensification and other strategies to maximise yields and revenues (Giampietro and Mayumi 2009). Consequently, monitoring this life cycle indicator is of importance in a country such as Luxembourg, where the substitution of traditional crop systems is being considered to enhance biocrops for energy production (Brandão et al. 2011). In fact, the data obtained in this study, despite failing to provide precise results on how bioenergy crops would affect the system concerning land use impacts, allows the identification of areas in Luxembourg that are prone to episodes of high erosion when changes in land use are performed. In addition, the crop allocation maps, calculated through transition probabilities, allow identification of vulnerable areas that coincide with areas of probable expansion of energy crops.

The second indicator that was analysed in the frame of the LANCA® model, PF, constitutes an important predictor of soil quality, especially when considering the potential for pollutant sequestration (Kai-hua et al. 2011; Tang et al. 2009), namely heavy metals, such as copper, lead or zinc (Arias et al. 2005). The reference values reported for the three land use types monitored in Luxembourg (i.e., forest, grassland and cropland) correspond to that of loamy soils, which usually range from 15 to 20 cmol $kg^{-1}{}_{soil}$ (Kai-hua et al. 2011). Considering that according to Schanne and Mathieu (2006), 44.4 % of land in Luxembourg can be assigned to loamy soils, the average results seem plausible. Moreover, the transformation and occupation impacts shown in Fig. 7 suggest minor increases in CEC in the arable land after the relaxation time, which, despite the strong correlation between CEC and clay content, would be connected to the predicted increase in organic matter, a parameter that can also determine, to a certain extent, the physicochemical properties of soils (Kai-hua et al. 2011; Paz-González et al. 2000). Finally, it should be noted that future research should try to determine spatial differentiation for physicochemical properties in order to understand the differences between different areas in Luxembourg. For instance, it seems plausible to presume that areas with clay predominance (i.e., the Gutland region, southern and central Luxembourg), may have higher CEC values due to the high clay content of the soils, which would imply a considerable decrease in soil quality due to restricted drainage, limited structure and risk of soil compaction (Kai-hua et al. 2011).

[4] Parcels with crop data accounted for 84.9 % of arable land in 2009, which was the year with the lowest coverage. The highest coverage corresponds to 2006 (93.4 %).

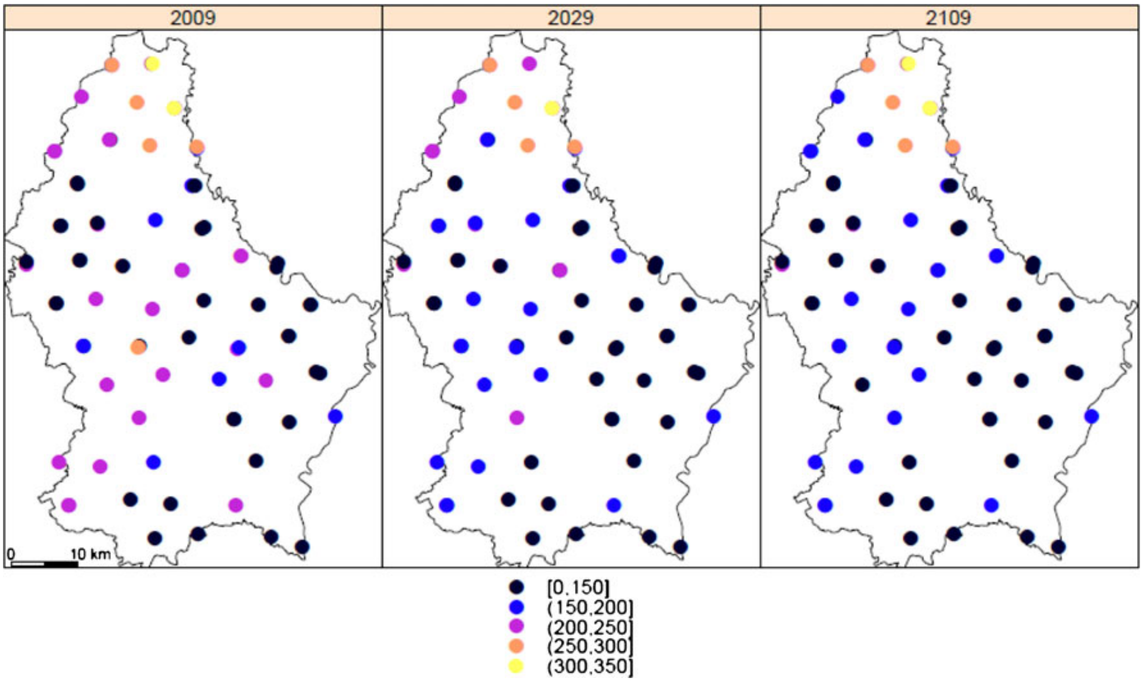

Fig. 10 Groundwater replenishment (GR) quality values for the scenarios selected. Note: results reported in mm/a

The other filtration capacity that is evaluated through LANCA® is MF, which is strongly dependent on the soil texture and on the groundwater distance of the analysed samples. Interestingly, it is the areas that show higher sand content in central Luxembourg, together with a high groundwater distance, that have the highest potential for water permeability. In addition, when analysing the occupation and transformation impacts, these areas in central Luxembourg

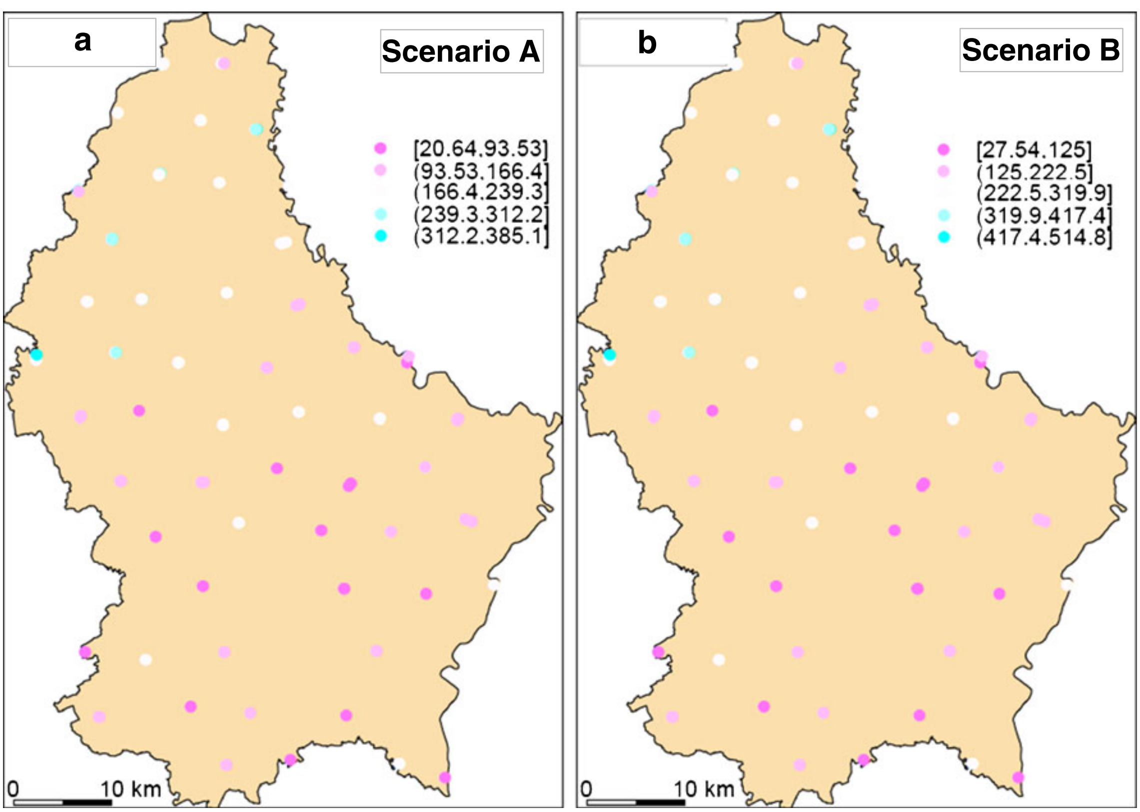

Fig. 11 Soil organic content (SOC) results for the scenarios selected. Note: results reported in t C/year

are also the areas with the highest potential for improvement after the theoretical restoration times.

BP in the LANCA® model is calculated based on the type of land use, without taking into consideration any possible parametrical differences between sample points (Beck et al. 2010). Therefore, in a similar way to the PF indicator, the data for this indicator cannot be spatially differentiated based on grids and can only be classified taking into consideration the land use in each area. Nevertheless, literature studies report important spatial differences in the net primary productivity (NPP) of crops, not only between different crop types but also as a function of other factors, such as management strategies, climatic conditions, leaf structures or canopy architectures (Peng et al. 2013). Therefore, an interesting future perspective to improve the feasibility of producing spatially differentiated results concerning BP may be the calculation of site-specific NPP values (Bolinder et al. 2007) or the combination of LCA and GIS coupling with remote sensing analysis (Wagendorp et al. 2006)

Finally, the GR indicator shows a clear distinction between the northern part of the country (Oesling region) and the Gutland region, where the benefits from shifting to restored natural landscapes would not be as visible in terms of soil quality recovery (see Fig. 10). The main carrier of this difference between the two regions is mainly the higher precipitations registered in the north of the country (up to 850 mm vs. 600 mm in the south—see Table S5 in the Electronic Supplementary Material).

4.3 Brief interpretation of the SOC indicator results

Results in Fig. 11 show how SOC values are strongly dependent on site-specific conditions. Nevertheless, the potential to retain additional amounts of organic carbon if land is allowed to restore to its natural state is more feasible in the north of the country, while the potential in southern areas is limited. In fact, this significant difference between the North and Central/South Luxembourg constitutes a regular trend throughout the assessed indicators, suggesting that the management of soil functionality in the two areas should be evaluated separately and attending to the specificities of each one. Moreover, it should be noted that the lack of primary data regarding the evolution of carbon content in Luxembourgish soils impedes a more precise interpretation of the SOC indicator. Finally, in a similar way as in the indicators used in the LANCA® model, results would benefit from the availability of more detailed crop rotation data rather than having to use broad land use types that constitute a barrier in consolidated land uses.

5 Conclusions

The results that were computed and analysed in this study allowed a better understanding of the main inventory drawbacks that were identified when applying the LANCA® model and the SOC indicator to the agricultural sector in Luxembourg. The main common limitation is linked to the fact that the LUC that are examined in Luxembourg are mainly associated with crop rotation rather than broader LUC types, which highly challenges the current barrier to obtain quality data to assess impacts related to soil functionality. Therefore, the limitations encountered in this study should stress the need to deepen the research regarding regional LUC, such as crop rotation. This methodological improvement would be of special interest in countries or regions such as Luxembourg in which the major LUC were made gradually ever since the arrival of the Industrial Revolution, and now have relatively stable global land use types due to legislation and other drivers.

Another important aspect highlighted in this article is the development of a consistent framework for the calculation of spatially differentiated land use impacts through the script realised in the R programming environment. Besides offering the possibility of an automatic spatially differentiated assessment of the land use impacts linked to any specific land coverage, this approach has the advantage of permitting an easy and fast update of the calculation every time new updated values (e.g., coming from new local measuring campaigns) are available for the input parameters. Moreover, the results acquired in the different indicators also allow a first approximation of the life cycle impacts affecting pedology and soil functionality in Luxembourg, providing a mechanism that may be of interest in policy-making in order to understand future challenges in terms of agricultural development, natural land conservation or global warming.

The inclusion of spatial predictions regarding crop occupation and rotation based on transition probabilities also provides a useful framework for understanding the effects of agricultural policies on the territory, on the landscape and, ultimately, on environmental impacts, including the ones assessed in this case study. Nevertheless, future research should focus on improving the rationale behind the transitions themselves in order to create integrated matrices that not only limit the prediction to annual crop rotation but also may also assimilate other important agricultural and behavioural aspects (i.e., soil texture, slope or farmer effect).

Finally, an interesting future development would be to provide a consequential rationale to the land use impacts obtained in this study in order to understand how different agricultural policies, including enhancement of biocrops production, may influence the marginal land use impacts occurring due to changes in the main production system.

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